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Eradication—preventing invasions at the outset

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A growing number of invasive animal populations—both vertebrate and invertebrate—have been completely eradicated. These projects usually have been on islands, but some have been on large continental areas, and many technologies have been used. Total eradication of plant populations has been reported less frequently, but there are nevertheless many successes. Though biological features may tend to make plant eradication more difficult than animal eradication, the difference in the number of success stories is probably more due to greater enthusiasm, persistence, and perhaps resources devoted to animal eradication than to biological differences. There is every reason to believe that many plant populations could be eradicated, particularly if eradication campaigns were coupled with a monitoring system that detects invasions early. Features conducive to successful eradication are (1) resources must be adequate, and there must be a commitment to see the project through to completion; (2) clear lines of authority must be established; (3) the biology of the species must be appropriate; (4) the target species must be detectable at low densities; and (5) subsequent intensive management of the system, such as for restoration, may be necessary. For success in some eradication campaigns, rapid reinvasion must be unlikely; in other instances the economics of the situation may make the attempt worthwhile even if reinvasion ensues. A failed eradication effort need not be a mistake, particularly if the eradication method used would have been utilized in traditional maintenance management. Further, the inspirational value of an eradication campaign and its enlistment of citizen support may help sensitize the public to the entire problem of invasive introduced species. Expanded eradication efforts can potentially effect enormous ecological and economic savings.

Key words: Benefit-cost analysis, eradication, islands.

Eradication is often a stepchild in the field of introduced species management (Simberloff 2002a, 2002b, 2002c). Rather, maintenance management is usually seen as the appropriate response—that is, controlling an invader at a density sufficiently low that we can tolerate it. Maintenance options are typically seen as mechanical, chemical, and biological control, plus ecosystem management (Simberloff 2002a). Politicians occasionally demand eradication of an invader, but removal of every single individual is a controversial goal (e.g., Myers et al. 1998), and scientists are skeptical (e.g., Dahlsten 1986) for three main reasons: eradication is not believed to be feasible, it may be costly, and it may entail collateral damage. Legendary failed eradications highlight these problems. An infamous failure was the 14-yr attempt by the United States Department of Agriculture (USDA) to eradicate the imported fire ant (*Solenopsis invicta*) in the southeastern United States (Davidson and Stone 1989), a fiasco in terms of collateral damage (including to human health and nontarget insects) and expense (over \$200 million) termed “the Vietnam of entomology” by E. O. Wilson (Brody 1975). This campaign probably exacerbated the fire ant invasion by causing greater mortality for its natural enemies than for the fire ant itself. The biology of the ant should have suggested that total elimination over a wide area was impractical.

However, many animal invaders have been successfully eradicated (Myers et al. 2000; Simberloff 2002b, 2002c), beginning with the tse-tse fly (*Glossina* spp.) from the 126-km² island of Principe in the Gulf of Guinea in the early 20th century (Lapeyssonie 1988). Many successful eradications have occurred on islands, ranging from small islands

(e.g., screw-worm fly [*Cochliomyia hominivorax*] from Curaçao [Baumhover et al. 1955], Asian citrus blackfly [*Aleurocanthus woglumi*] from Key West [Hoelmer and Grace 1989], Oriental fruit fly [*Dacus dorsalis*] from Rota and Guam [Steiner and Lee 1955; Steiner et al. 1965, 1970], Pacific rats [*Rattus exulans*] from Tiritiri Matanga [Veitch and Henry 2001]) to very large ones (e.g., nutria [*Myocaster coypus*] from Great Britain [Gosling 1989], melon fly [*Bactrocera cucurbitae*] from the entire Ryukyu Archipelago, including Okinawa [Iwahashi 1996; Kuba et al. 1996]).

Successful eradication is not just an island phenomenon. The most widespread effort undertaken was the eradication of smallpox from the face of the earth (Fenner et al. 1988). The African mosquito, *Anopheles gambiae*, a vector of malaria, was eradicated from 31,000 km² of northeastern Brazil (Davis and Garcia 1989; Soper and Wilson 1943). Other eradications from large parts of continents include the screw-worm, first from Florida, then from the southeastern United States, then from Mexico, and most recently from several Central American nations (Galvin and Wyss 1996; Reichard et al. 1992), and bovine contagious pleuropneumonia from the United States (Fenner et al. 1988). Eradication from smaller continental areas is fairly common, such as the elimination of the giant African snail (*Achatina fulica*) from a region of south Florida (Mead 1979) and part of Queensland, Australia (Colman 1978), the medfly (*Ceratitis capitata*) from 20 Florida counties (Simberloff 1997a), and yellow fever from Panama (Fenner et al. 1988).

Plants are not well represented among this tally of eradication successes. However, witchweed [*Striga asiatica* (L.) Ktze.] eradication in the Carolinas is a notable project head-

ing toward success (Eplee 2001; Westbrooks 1993); I will discuss this effort later. Karoo thorn (*Acacia karoo* Hayne) has been eradicated from Western Australia and Victoria, and Taurian thistle (*Onopordum tauricum*) from Victoria (R. Groves, personal communication; Weiss 1999). Of the seven plant eradication projects in addition to witchweed sponsored by the USDA Animal and Plant Health Inspection Service through 1993, only the elimination of Asian common rice (*Oryza rufipogon* Griff.) in 0.1 ha of Everglades National Park appears successful, although the ranges and densities of some of the other targets, such as branched broomrape (*Orobanche ramosa* L.) and goatsrue (*Galega officinalis* L.), were substantially reduced locally (Westbrooks 1993). In Kruger National Park, South Africa, 10 invasive plant species have been eliminated (Macdonald 1988).

Of course, in addition to famous failures such as the fire ant campaign, many other attempted eradications have not completely eliminated the invader; surely there are more such cases than total successes. I have not attempted a tally because the literature is too scattered and uncertain and because colloquial use of the term "eradication" makes it difficult to assess exactly what is a failure (Simberloff 1997a, 2002b). Often, public figures (e.g., Chiles 1996) and even scientists (e.g., Langland and Sutton 1992) use "eradication" to mean partial removal and substantial control. In these instances there was no real attempt to eradicate. Should such a campaign be viewed as a failure? This assessment seems unduly harsh if the same method has been used for maintenance management as would have been used for true eradication, particularly if substantial control results, as in the attempt to eradicate *Spartina* spp. from New Zealand (Nicholls 1998). For example, in Scotland, alien giant hogweed (*Heracleum mantegazzianum* Somm. & Lev.) has been well controlled within the Aberlady Bay Local Nature Reserve by cutting and selective herbicidal spraying (Usher 1973). There is continued reinvasion, so that true eradication, the stated goal, is at best ephemeral, but it seems draconian to term the campaign a failure because normal maintenance management would have used exactly the same methods as did the eradication campaign.

This article examines successful and unsuccessful eradication campaigns to see whether common features characterize successes. Under what circumstances is eradication feasible? Is plant eradication inherently less feasible than animal eradication? I emphasize that I am not asking whether society as a whole wants a particular invader removed or even managed. This is sometimes a contentious issue. A well-known example is the fight in Australia over *Echium plantagineum* L., termed Paterson's curse by ranchers and Salvation Jane by apiarists (Cullen and Delfosse 1985). Rather, I will assume that society does want to control a particular species; the question is how.

Economic Resources

Very small-scale eradication need not require enormous resources; the drive and dedication of a single person or a small nongovernmental organization may even suffice. For instance, a devoted group of scientists (the Island Conservation & Ecology Group) has removed various combinations of feral cats (*Felis catus*), Norway and black rats (*R. norvegicus* and *R. rattus*, respectively), house mice (*Mus mus-*

culus), rabbits (*Oryctolagus cuniculus*), goats (*Capra hircus*), sheep (*Ovis aries*), and burros (*Equus asinus*) from nine islands in northwest Mexico (Donlan et al. 2000). However, for large areas, costs are often huge. For 50 infestations of 16 plant pests in California, Rejmánek and Pitcairn (2002) found that log (of cost) increased linearly and rapidly with log (of infested area). Successful large regional eradications have been supported by significant government resources or private investment. The Rockefeller Foundation and the Brazilian government funded the Brazilian eradication of *A. gambiae* (Davis and Garcia 1989), United States taxpayers paid \$750 million for the screw-worm eradication in the United States and Mexico (Reichard et al. 1992), and the strong support of the United States government as well as the state governments of North and South Carolina has enabled the reduction of the African root parasite, witchweed, in the Carolinas from infestation of 162,000 ha in the 1950s to ca. 2,800 ha currently (Eplee 2001; Westbrooks 1993). Of course, huge budgets do not ensure success, as witnessed by the fire ant eradication disaster. But for large areas, substantial funding is usually a prerequisite (Myers et al. 2000; Simberloff 2002b).

The rapid increase in expense as area of an invasion increases leads to the dictum that it is best to eradicate early (e.g., Myers et al. 2000; Simberloff 1997a; Weiss 1999). Although some longstanding, widespread invasions have been eradicated, likelihood of success is obviously improved and cost minimized if an invasion is nipped in the bud. This fact, of course, argues for effective early-warning and rapid-response machinery (Simberloff 1997b; Weiss 1999), a subject beyond the scope of this paper. A case that exemplifies the benefits of quick action if eradication is the goal is the "killer alga," *Caulerpa taxifolia*. This tropical alga almost surely could have been eliminated in the Mediterranean within a few years of its discovery when it inhabited but a few square meters in front of the Oceanographic Museum of Monaco. However, the effort was delayed, and the alga now infests several 1,000 ha of the coasts of Spain, France, Monaco, Italy, Croatia, and Tunisia (Meinesz 2001). On the other hand, an effort to eradicate a small infestation of the same alga near San Diego within a year of its discovery seems promising (Meinesz 2001). An attempt, using similar methods on a much larger (and older) infestation near Los Angeles is less promising.

Some substantial weed infestations in the United States could probably have been eradicated had an attempt been mounted soon after discovery. For instance, common crupina (*Crupina vulgaris* Cass.) was first detected on a mere 18 ha in Idaho in 1969 (Stickney 1972). By 1981, it infested 9,000 ha and was listed as a Federal Noxious Weed (Westbrooks et al. 2000). An eradication feasibility study was launched that year, and Thill and colleagues (e.g., Miller and Thill 1983; Thill et al. 1985; Zamora and Thill 1989) provided extensive biological data suggesting that total elimination was possible. They included description of a successful elimination of an isolated infestation of 0.8 ha in California. However, the feasibility study was not completed until 1988, and although it determined that eradication was feasible, a federal-state task force to plan the project did not convene until 1991 (Westbrooks et al. 2000). Common crupina had by then reached California, Oregon, and Washington as well as Idaho, and it dominated 25,000 ha. The

task force finally decided against immediate action on the ground that the necessary herbicide might have harmful effects on salmon.

As a contrast, a project at modest cost (so far ca. \$250,000) in Western Australia to eradicate kochia [*Kochia scoparia* (L.) Schrad.] (also called summer-cypress) is nearing success (R. Randall, personal communication; Randall 2001). Introduced in 1990 as a salt-tolerant forage, kochia was planted on 52 properties. However, by 1992, it was seen as a weed, and an eradication campaign using herbicides was mounted. By 1993 it had established on 270 properties spread out across a linear distance of over 900 km, with more than 3,200 ha infested. By 1995, the infested area was reduced to 139 ha and by 2000, to 5 ha.

Certain eradication expenses can be substantial and surprising (Myers et al. 1998; Simberloff 2002c). Killing the first 99% of a target population can cost less than eliminating the last 1%. Government agencies may not understand this and may decrease funding once the problem subsides, rather than seeing it through to completion (Mack and Lonsdale 2002). This has been a persistent problem in management of hydrilla [*Hydrilla verticillata* (L. f.) Royle] in Florida (Schardt 1997). Costs of monitoring may increase when pest densities are very low, yet intensive monitoring is the only effective way to determine when to end an eradication campaign. In some eradication projects, an expensive public relations campaign may be needed to ensure public support and avert lawsuits (Myers et al. 1998). For instance, for just part of the California medfly (*C. capitata*) eradication project, the state of California paid \$3.7 million to settle 14,000 claims for damage to car paint (Getz 1989).

Lines of Authority

It is hard to persuade large groups of people with diverse interests to support an eradication campaign whose benefits are not seen as equal to all. Because eradication can be subverted by one or a few individuals, a government agency or interagency entity must be able to compel cooperation (Myers et al. 2000; Simberloff 2002b). Where there is strong distrust of government, this authority will generate opposition (cf. Perkins 1989). Objections to eradication techniques may be so substantial that only a strong governmental authority can enact the program. Aerial spraying of malathion to eradicate medflies fostered widespread complaints about discomfort or threats to human health in California (Penrose 1996) and Florida (Anonymous 1997), and killing of large vertebrates by trapping, hunting, or poisoning often generates vocal opposition (e.g., feral pigs [*Sus scrofa*] in the Hawaiian islands [J. Van Driesche and R. Van Driesche 2000], nutria in Great Britain [Gosling 1989], monk parakeet [*Myiopsitta monachus*] in the United States [Simberloff 1997a]). Even invasive plants can have enthusiastic supporters who impede control efforts. The destruction of diseased Japanese cherry trees (*Prunus serrulata* Lindl.), presented by the city of Tokyo in 1909 to its sister capital Washington, was denounced as xenophobic (Pauly 1996); the eradication of invasive *Eucalyptus* trees from Angel Island, California, gave rise to charges of brutality and "eucalyptus phobia" (Azevedo 1990); and the massive removal of invasive *Casuarina equisetifolia* L. ex J. R. & G. Forst. in south Florida fostered many objections (e.g., Gasco 1995)

on grounds ranging from aesthetic and ecological to that of public health (U. Celmins, personal communication).

When human health is at stake, as in the tse-tse eradication in Nigeria (Oladunmade et al. 1986) or the malaria mosquito eradication in Brazil, even draconian government control is less likely to generate opposition. When an eradication campaign benefits agriculture, while the entire public bears the costs and possible side effects, as in spraying malathion to kill medflies, conflict is more likely (Simberloff 2002b, 2002c). The recent attempt to stem citrus canker in Florida also has generated enormous opposition (Hiaasen 2002; Vogel 2001), extending even to death threats (Sharp 2000). Most eradications attempted for ecological or conservation purposes have occurred on small islands or in reserves, often with little or no human population, and opposition has been rare. Until conservation is more highly valued by the whole citizenry, I predict that attempts to eradicate ecological pests over wide areas will generate hostility because of either economic costs or side effects. On a small scale, local attempts to eradicate Asian long-horned beetles (*Anoplophora glabripennis*) by felling urban trees in Chicago and New York gave a foretaste of the complaints that may arise if this campaign has to be greatly extended (e.g., Stout 1996; Toy 1999); of course, the ultimate purpose in this instance is silvicultural more than ecological. I know of no large-scale eradication projects conducted solely for ecological reasons.

Biology of the Target Species

With sufficient resources, it is probably possible to eradicate almost any species in a small area, although certain biological features can make a target less tractable. However, the biology of the target species may be crucial, and scientific knowledge must be profound (Fenner et al. 1988; Myers et al. 2000; Simberloff 2002b). Some traits obviously make eradication easier—for example, large mammals are far easier to find than small insects; large size probably also makes plants easier targets on average, as can be seen by the fact that eradication campaigns in Kruger National Park were successful for 28% of 25 species of trees and shrubs but only 3.4% of 88 plant species of other growth forms (Macdonald 1988). Plants with a soil seed bank are more difficult than are those without this feature (Simberloff 2002b). However, key biological points often require substantial research, especially in natural history. For example, among successfully eradicated species, smallpox has no non-human reservoir or long-term carriers (Fenner et al. 1988), the giant African snail does not self-fertilize (Mead 1979), *A. gambiae* in Brazil was found almost exclusively near buildings (Davis and Garcia 1989), and citrus canker (caused by *Xanthomonas axonopodis* pathovar *citri*), eradicated in the southeastern United States in the early 20th century, had a highly restricted host range and spread only by human movement of infected hosts (Merrill 1989).

Plants as a group may present fewer prospects for successful eradication than do animals, but it still seems likely that eradication would be a feasible goal for many plant species. As with animals, some types of plants (e.g., trees) are easier to eradicate than others (e.g., herbs), and again, as with animals, certain biological features (e.g., a soil seed bank, various long-distance dispersal mechanisms) make

eradication of some species more difficult. It is possible that successful eradication of plants would, on average, take a longer campaign than would eradication of animals. For instance, exhaustion of soil seed banks would require a persistent effort. Probably, a bigger advantage that animal eradication often has over plant eradication is that animal behavior allows the possibility of attracting the target species, as in the Judas goat technique (allowing a radio-collared individual to join the wild herd [Parkes et al. 1994]), the male annihilation method for insect eradication (Steiner et al. 1965), or the use of poison baits for various mammals.

Probability of Reinvasion

Is it worth eradicating an invader if rapid reinvasion is likely? One reason so many eradication attempts have been made on islands is that rapid reinvasion is less likely. In many circumstances, a successful eradication campaign is counterbalanced by reinvasion. An intensive campaign in Washington state eliminated Eurasian watermilfoil (*Myriophyllum spicatum* L.) from the 130-ha Long Lake (Thurston County Department of Water and Waste Management 1995), but a public boat ramp ensured rapid reinvasion, and the county switched to a program of maintenance management by hand-pulling (M. Swartout, personal communication). The probability of deliberate subversion of an eradication (Perkins 1989) is so high that the attempt may be futile. In the Bay area of California, an individual known as Johnny Weedseed is suspected of planting South African capeweed [*Arctotheca calendula* (L.) Levyns] and other rampant exotics in natural areas such as Golden Gate National Park (Davis 1990). The ease with which an individual can “salt” an area with small plants may make it difficult to prevent reinvasion.

In general, whether the likelihood of reinvasion should argue against an eradication campaign rests on a benefit-cost analysis. Even if reinvasion is probable, eradication may be a rational decision so long as the target pest remains absent from the area in most years. For instance, the benefit of an eradication campaign may be biologically artificial—trade regulations may prohibit importing some product unless its region of origin is certified as free of pests. The economic benefits could then mean that even with certain reinvasion, eradication would be a rational choice. This is why costly eradication campaigns are repeatedly mounted against the medfly in California and the gypsy moth (*Lymantria dispar*) in parts of the United States and Canada despite likely rapid reinfestation (Myers et al. 2000). I do not mean to imply that the ecological and economic benefits of either campaign may not suffice to justify them even in the absence of trade regulations. Rather, the important point is that maintenance management rather than eradication is not an option because of trade regulations, even if maintenance management would achieve equivalent or greater real control or cost less.

Independently of trade regulations, an eradication campaign can have sufficient economic, ecological, health, or even symbolic benefits to warrant the cost even if quick reinvasion is certain. Often mechanical removal of plants is colloquially termed “eradication,” and it can be effective (but very labor intensive) even if reinvasion is automatic. Conservation organizations such as The Nature Conservancy often use volunteer labor (e.g., Randall et al. 1997). In

Florida, “the Pepper Busters” volunteer program has been crucial to attempts to control the state’s worst invader, Brazilian pepper (*Schinus terebinthifolius* Raddi) (Zarillo 1999). Many volunteer efforts, by employing large numbers of citizens, arouse the public and engage them in the battle against invasive exotics at the same time as they provide a measure of control. In Victoria, British Columbia, Canada, the Garry Oak Meadow Invasive Plant Removal Project, with many “broom bashes” to remove Scotch broom [*Cytisus scoparius* (L.) Link], has engaged so many citizens that a monthly listing is required in the local environmental newsletter (Econews 1998). The campaign produces much local publicity (e.g., Curtis 1996) about both Scotch broom and exotic plants in general. Most importantly, it enlists many young people, such as elementary school children and Girl Guides (V. Nealis, personal communication), in the battle against exotic weeds. Although all the above projects are fundamentally aimed at maintenance management, such labor-intensive efforts could be mobilized to aid permanent eradication projects.

Possibility of Restoration

Eradication as part of an ecological restoration project may be defeated by reinvasion or other problems (Simberloff 2002b). Just removing an exotic species does not constitute restoration (Simberloff et al. 1999; Towns et al. 1997). Prominent species originally present may now be extinct, and no acceptable functional equivalents may be available. Some restoration efforts fail mysteriously. For instance, after eradication of predators, stitchbird (*Notiomystis cincta*) reintroduction to New Zealand islands has failed for no obvious reason (Towns et al. 1997). Our knowledge of community structure and function is inadequate to predict with assurance the effects of removing a prominent member of an ecological community. Thus, unforeseen effects of eradication abound (Towns et al. 1997). For instance, removal of an introduced herbivore can lead to proliferation of exotic weeds rather than restoration of the native plant community. Eradication of rabbits from Motunau Island led to increases of exotic boxthorn (*Lycium ferocissimum* Miers) (Towns et al. 1997), and removal of grazing livestock from Santa Cruz Island (California) caused dramatic increases in fennel (*Foeniculum vulgare* Mill.) and other nonnative plants (Dash and Gliessman 1994). Some of these effects of eradication might have been predicted, but others are so mysterious that substantial scientific research has failed to suggest a reason; in short, eradication is often a large, uncontrolled experiment, and we should expect surprises (Simberloff 2002b, 2002c).

Economics of Eradication

I have discussed whether eradication is feasible, with some attention to benefits that might arise even from unsuccessful eradication projects, as well as to unforeseen problems. I have dodged the matter of whether eradication is an appropriate strategy even if it is feasible. The prospect of permanent elimination of an invader from a site or region, and thus elimination of annual management costs as well as the danger of some delayed effect of the invader (fairly common among introduced species; see Crooks and Soulé 1996), is alluring. However, given the costs that successful eradication

may entail, society cannot undertake to eradicate every pestiferous invader for which eradication is feasible. When is eradication the best management option? Such decisions are generally based on benefit-cost analyses (Arrow et al. 1996), but benefit-cost analyses of many natural resource issues, especially those related to conservation, are difficult because there is often no market for an environmental or conservation resource as there is for an agricultural commodity (LeVein 1989; Simberloff 1992). In the new field of invasion economics, benefit-cost analyses are especially problematic and perhaps have never been adequately conducted (Perrings et al. 2000). One problem is that it is very hard to predict the trajectory of invasions, and another is that it is very difficult to predict the effects of various management measures. Benefit-cost analyses will have extremely wide confidence limits for many years to come.

However, in some circumstances, it seems obvious that an eradication attempt would be justified by a comprehensive benefit-cost analysis. For smallpox (Fenner et al. 1988), the entire annual national and international cost of the eradication, from the inception of a full-fledged campaign in 1967 to its success in 1979, was only \$23 million, whereas the annual cost of the disease during this period in underdeveloped nations alone was at least \$1.07 billion, and worldwide it was estimated at \$1.35 billion. The annual cost of control efforts in the United States alone before the eradication campaign was \$150.2 million. Even if the campaign had not succeeded, so long as it had even a moderate probability of success, it would seem to have been an appropriate investment. Unfortunately, benefit-cost analyses for weed management are rarely so clear-cut (e.g., Simberloff and Stiling 1998).

A scan of the annual current management costs (not including losses and damages) estimated for some invaders in the United States alone (Pimentel et al. 2000) suggests that even an expensive eradication project might be justified, so long as prospects for success were even moderate and the attempt would not preclude other effective management techniques. Every year, the United States spends \$45 million on purple loosestrife (*Lythrum salicaria* L.) control, \$3 million to \$6 million on management of melaleuca [*Melaleuca quinquenervia* (Cav.) Blake], and \$100 million to deal with Dutch elm disease (*Ophiostoma ulmi*). However, the real prospects of successful eradication of any of these species, along with the likelihood of nontarget effects, would have to be assessed based on detailed knowledge of its biology, and it could be that alternative methods (e.g., the recently released biological control agents for the first two species) may produce adequate control and acceptably low nontarget effects at far lower than the current costs.

I list these examples to show that each one entails a huge annual expenditure, and I wonder if the possibility has been considered that total, long-lasting eradication could be achieved for, let us say, 10 or 20 times the current annual control cost. The idea has often been rejected out of hand (e.g. for purple loosestrife [Young 1989]). Do administrators of our natural resources typically think this big? I also wonder if persistent, massive mechanical control is really being considered. In addition to the great amount of available volunteer labor noted above, in the United States there is a large, growing convict labor pool that can potentially assist in eradication projects. Florida inmates are already a crucial

component of successful efforts to reduce the area occupied by melaleuca (Campbell and Carter 1999). Perhaps they could help to get rid of it altogether. In Kentucky, the State Nature Preserves Commission has had good success in managing musk thistle (*Carduus nutans* L.) in certain areas by using volunteers who have been convicted of driving under the influence of alcohol (J. Bender, personal communication). Of course, paid labor is also an option when society recognizes that removal of introduced species is worth the expense. In South Africa, the Working for Water Programme, a large public works project, has played a key role in battling damaging exotic plants by employing teams of people to remove infestations manually (McQueen et al. 2000; van Wilgen et al. 2000). With persistence, even substantial seed banks in the soil can be exhausted, as with Brazilian *Araujia sericifera* Brot. (moth plant) on the Poor Knights Islands, New Zealand (Coulston 2002).

Conclusions

Eradication sometimes succeeds even over large areas. And there is a growing string of smaller successes, though many more are with animals than with plants. However, eradication remains almost a stepchild of management of invasive species, often omitted while considering options for dealing with invasions, even when the details of an invasion might augur well for success. I do not believe that the relative paucity of plant eradications is primarily due to inherent biological differences in various traits relevant to eradication, even though such differences exist. Rather, I see two main reasons for the low visibility of eradication in general and plant eradication in particular.

First, the eradication literature is scattered and often very gray. Vertebrate eradication is published in different outlets from those of insect eradication, and plant eradication histories are found in yet other sources. Sometimes, eradication histories are passed by word of mouth. What is needed is for managers of eradication projects to see high-quality international publications as a routine part of the job; even when an eradication attempt fails, publication is warranted. For eradication to become a frequent option in managing invasive species, methods and results have to be publicized better.

Second, the whole problem of introduced species seems so overwhelming that it has led to defeatism—the forces causing invasion, especially the growing movement of cargo and people in the free-trade era, seem so overwhelming that some authors think that we are doomed to global homogenization (e.g., the “planet of weeds” of Quammen 1998). Eradication, both because of its frequently bad press and because it is the management approach that aims the highest, falls victim to this defeatism more acutely than do other approaches. However, this sense of doom is unwarranted. We know that eradication can succeed because it has. It has sometimes succeeded despite the poor lines of communication I have just noted and despite the formidable biological powers of the target invader. We do not know the limits of eradication technologies. With enough manpower, just how large an island could be cleared of a dominant weed? If political will and economic support could be mustered, could purple loosestrife be completely eradicated from North America, and *Melaleuca* from Florida? Under what

circumstances could the witchweed approach be replicated? If smallpox and citrus canker can be eradicated, are plants on continents really out of the question?

I cannot answer these questions, but the successes in the literature should inspire us to think big. Just before the successful eradication of smallpox from the earth, the renowned scientist René Dubos (1965, pp. 378–379) wrote: “. . . it is easy to write laws for compulsory vaccination against smallpox, but in most parts of the world people would rather buy the vaccination certificate than take the vaccine; and they shall always find physicians willing to satisfy their request for a small fee. . . . For this reason, and many others, eradication programs will eventually become a curiosity item on library shelves, just as have all social utopias.” One thing is certain—we will surely become a planet of weeds if we do not aim high!

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